II. " A"FIRST EXERCISE IN ASSESSING THE BENEFITS OF CONTROLLING ACID PRECIPITATION

Introduction

In this chapter we undertake a first exercise in assessing the economic impacts of acid precipitation or acidifying deposition, given that it occurs at above-background levels. The main benefit of the construction, from the authors' perspective, has been to serve as a learning and organizing device about the state of natural science knowledge of acid precipitation effects upon life and property. We have concluded that the state of this knowledge is very incomplete, both in terms of empirically testable propositions derived from a broadly encompassing analytical structure as well as in quantitative bits of information that have been related to or associated with each other. Under these circumstances, it is tempting for the economist to plead the near-impossibility of his task, as if the natural scientist were responsible for any failings of the economist's attempts to value the effects. On some occasions, the plea is valid. On this occasion, 'it is, on balance, invalid. The reasons are two.

First, whatever the available research time and resources, economic analysis generally does not yet know how to assess quantitatively disruptions in ecosystem functions having major and broad economic impacts. [Building upon Scarf's (1973) work, Shoven and Whalley (1977), King (1980), and a few others show that the truth of this statement could be short-lived]. If the impacts of acid precipitation upon ecosystem nutrient storage, detrital decay, succession patterns, genetic pools, etc., are as dire as some natural scientists predict, and if alterations in these functions can legitimately be viewed as changes in natura 1 transformation processes (production technologies), then a full economic assessment may be analogous to comparing the welfare of the 18th century Jeffersonian yeoman farmer with his modern agricultural corporate clone: the worlds in which the two live(d) are so vastly different that neither the modern nor the Jeffersonian man could comprehend most of the opportunities and dangers familiar to the other. It is questionable whether the comparison would be economically meaningful.

The second reason why the blame for the economic limitations of the content that follows cannot be shifted to the natural scientist is because, in principle, it is both possible and practical to value many, perhaps most, of the effects of acid precipitation. The task is substantial but nevertheless accomplishable. Economists who read this chapter will recognize that only the most elementary economic analysis has been performed. In particular, we have generally resorted to an assumption throughout that acid precipitation affects only the yields from existing sets of economic activities. We, therefore, have, with only a few exceptions, disregarded any potential price effects, as well as changes in activity and location patterns. Finally, we have nothing to say for now about the economic implications of larger questions on changes in lifestyle, possibly unacceptable risks due to the destruction of life support systems, and the welfare of future generations. Effects on these larger questions, as well as those involving changes in activity and location patterns, are most likely to be generated by the buffering stock depletion impacts of acid precipitation. It is these stock depletion effects that, as was indicated in Chapter 1, economically distinguish acid precipitation effects from traditional analyses of pollution effects. In a later theoretical chapter, we will view these stock depletion effects as analogous to drawdowns in the biogeochemical energy available to a geographic location. In this chapter, so as to remind the reader that we do not view effects on the yields of current economic activities as the sole effect of acid precipitation worthy of economic attention, we present a brief treatment of the impact of acid precipitation on the buffering capacities of natural ecosystems.

Depletion of the Stock of Buffering Capacity

After deposition, the effects of acidifying components depend on sensitivities of the environments where the deposition occurs. This sensitivity is largely determined by the abilities of the depositional surface to buffer hydrogen ion additions. In turn, environmental buffering abilities initially depend on the bedrock and geological. history of the region. Bedrock of volcanic or igneous origin tend to be low in most minerals important in buffering [Dillion and Kirchner, (1975)]. Over geologic time, the importance of the bedrock is moderated by glaciation, weathering and other soil building processes.

Buffering in soil systems is primarily accomplished through cation exchanges between soil solutions and colloidal clay and humus particles, also called micelles [Buckman and Brady, (1960)]. These micelles are negatively charged and, therefore, attract positively charged cations which enter soil solutions during soil mineralizati.on, for example. Cations which are commonly adsorbed onto micelles in order of increasing affinity for micelle adsorption are: sodium, potassium, magnesium, calcium, aluminum and hydrogen. Increasing concentrations of hydrogen ions in soil solutions, as would occur with incident acidifying depositions, shift the equilibrium between the soil solution and the micelles such that additional hydrogen ions become adsorbed onto micelles, thus displacing cations having less affinity. Adding lime to soils causes concentrations of hydrogen. ions in soil solutions to decrease as they react with the calcium ions. This shifts the equilibrium between soil solution and micelles: hydrogen ions desorb and calcium ions adsorb onto micelles.

The second major environmental buffering system, the primary buffering mechanism for aquatic environments, is the carbonate-bicarbonate system. While this system is also present in soil environments, it is generally of relatively minor importance [Buckman and Brady, (1960)]. In aquatic environments, the abilities of the carbonate-bicarbonate buffering system are normally measured by alkalinity determinations [Sawyer and McCarty, (1967)]. Buffering capabilities develop as carbon dioxide dissolves in water. The carbon dioxide reacts with water to form carbonic acid which dissociates to form bicarbonates and hydrogen ions. Bicarbonates can then further dissociate to carbonate and hydrogen ions. Additions of hydrogen ions to the aquatic environment will reverse this process and, with continued additions, carbon dioxide can eventually be released from the water.

Continued deposition of acidifying substances in ecosystems will cause buffering capacities to decrease as exchangeable ions, carbonates, bicarbonates and, eventually, pH values decrease. Simultaneously, titratable acidity, hydrogen ion concentrations and associated anions such as sulfates and nitrates increase. Also, mineralization rates of soil particles will increase with hydrogen ion additions and counter, at least for short periods, the effects of acidification [Maimer, (1.976)].

The frequencies and durations of acidifying depositional events will affect the severity and rate of ecosystem changes. However, as eposodic acid contributions continue, a system's buffering capacity is continually reduced and its hydrogen ion concentration increased. As the pH continues to decrease, the impacts on the ecosystem increase and the importance of the episodic frequency of the acidifying contributions in defining ecosystem impacts decreases.

Sufficient additions of acidic water to soil systems can cause the cation released through ion exchange buffering to leach from the system. The decreased pH can have additional effects on soil chemistry [Buchman and Brady, (1960), Maimer, (1976)]. As soil pH decreases below 8.0, the amounts of aluminum, iron, and manganese increase in soil solutions. At low pH levels, concentrations of these compounds can become toxic; and as concentrations of these dissolved compounds increase, they can react to fix phosphates as insoluble hydroxyl-phosphates. In such a complex this valuable nutrient is not available for plant use.

Acidifying depositions enter aquatic ecosystems directly through surface deposition and indirectly through watershed runoff. Increases in the acidity of rivers and lakes occurs with acid precipitation events. But, depending on the buffering ability of the water, such changes may be relatively small and short in duration. More substantial acidity increases of longer duration can accompany spring snowmelt. These changes can be particularly dramatic when the snowpack melts rapidly [Likens et al., (1977)]. At such times, the pH of surface waters can decline to about 3.0 [Shaw, (1979)]. Gjessing et al. (1976) noted that when spring meltwaters are less dense than lake waters (water has a maximum density at $4^{o}C$), the meltwaters tend to flow over the surface waters of the lake and therefore do not mix with lake waters. During such times, waters having elevated acidities are discharged from lakes to produce maximum impacts on stream ecosystems. Runoff to lakes at other times of the vear generally mixes with lake waters, lessening the impacts on streams.

The influences of acidifying depositions on the chemistries of natural waters are similar to the effects produced on soils. Increasing the hydrogen ion concentration reduces the aquatic system's buffering abilities, increases solubilities of metals, complexes phosphates, etc. Continued addition of hydrogen ions causes carbonates and bicarbonates to be converted to carbon dioxide and water [Lewis and Grant, (1979)]. Carbon dioxide can then be lost to the atmosphere or dissolved concentrations can accumulate to levels which are directly lethal to aquatic organisms [EIFAC, (1969)]. Precipitation of normal suspended silt loads has also been noted for streams having increased hydrogen ion loading rates [Parsons, (1965)].

Studies of effects from acidifying depositions have shown the importance of buffering capacities in determining the constituents flushed out of watersheds. Often the input of hydrogen ions is adsorbed by the watershed ecosystem and no significant increase in hydrogen ion or other cation concentrations is observed in aquatic system outputs (e.g., Lewis and Grant, 1979). In other cases, the uptake of hydrogen ions by the watershed results in an increase in the output of other cations from the watershed. For example, Gjessing et al. (1976) report that outputs of calcium, magnesium, and aluminum ions from nine watersheds in Norway were proportional to inputs of hydrogen ions. In contrast, Lewis and Grant (1979) noted no significant change in outputs of calcium, magnesium, sodium, potassium, phosphate, or hydrogen ions accompanying increased hydrogen ion inputs to a Colorado, USA, watershed. Increased outputs were observed for sulfate, nitrate, ammonia and dissolved organic matter, while decreased outputs of bicarbonate were proportional to increased hydrogen ion inputs. Variations in output responses among the Norway and Colorado watersheds suggest that different buffering systems are responding to the hydrogen ion inputs. The Norway watersheds appear to buffer primarily through cation exchange, primarily a terrestrial system, while the Colorado watershed appears to buffer primarily through reactions with bicarbonate. As both the Norway and Colorado watersheds are of granitic origin, variations in the buffering systems may reflect the relatively longer time which Norway has been exposed to acidifying depositions. With continued additions of acids, bicarbonate buffering systems become exhausted causing buffering to become increasingly dependent on cation exchange.

Knowledge of buffering capacities is helpful in sorting out habitat impacts of hydrogen ion inputs from impacts of other chemical constituents associated with acidifying depositions. Taken together, variabilities in depositional composition and mode as well as receptor buffering can be used to discriminate among these effects. In poorly buffered systems, depositional impacts tend to be more related to pH effects because of the low masses of SO_x and NO_x required to generate pH changes. Tn well buffered systems, the masses of SO_x ‘and NO_x necessary to generate pH changes are relatively larger and, consequently, influences of other compounds tend to be enhanced. Thus, well buffered systems tend to respond more to components other than hydrogen ions in both wet and dry acidifying depositions; this relationship will tend to be maintained until the buffering capacity of the system is exhausted. With poorly buffered terrestrial systems, wet depositions will tend to have confounded responses to both hydrogen ion concentrations and other depositional contents until pH effects overwhelm other responses. Dry depositions to poorly buffered terrestrial systems will tend to cause responses primarily attributable to the depositional. compound (e.g., SO_x or NO_x); in some instances biochemical transformation and utilization of the compounds will generate accumulations of hydrogen ions causing pH effects to predominate eventually. Needless-to-say, poorly buffered aquatic systems will have similar responses to both wet and dry depositions as dry depositions essentially become wet deposition upon entrance into the aquatic system. These responses will be similar to wet depositions in poorly buffered terrestrial systems.

Agricultural Effects

Low-pH precipitation can affect crop yields in two ways. First, as it percolates through the soil column it accelerates the natural tendency of water to leach organic and mineral soil components from the root zone. At the same time, it reduces the soil pH level in this zone, thus making nutrients less available and toxic metals, such as soluble aluminum and iron, more available to plants. In addition, reduced soil pH levels can cause declines in microbe populations that break organic matter down into forms useful for plants. In the absence of this breaking down, the organic matter can accumulate and seal the upper layers of the surface while permitting various plant toxins to be formed from the matter. The result for all these impacts is reduced growth and yields for plants located on the acidified soils [Buchman and Brady (1960)]. According to Schwartz and Follett, (1979, p. 2) the "preferred soil pH range for maximum growth" varies between 5.5 and 7.0 for most commercially important crops, e.g., clover, barley, corn, grasses, and soybeans. However.. this is by no means universal across crops, since the interval for alfalfa is 6.2 - 7.5; for asparagus and lettuce, it is 6.0 - 7.0; for blueberries and sweet potatoes, it is 5.0 - 5.7; for white potatoes, it is 5.0 - 5.4; and for cotton, it is 5.5 - 6.5. In the agricultural regions east of the Mississippi River, soil acidity levels frequently fall below these "maximum growth" intervals because of the region's high rainfall and because of grower soil additions of inorganic fertilizers. Consequently, it is a standard grower practice calcitic or dolomitic ground limestone with the soil periodically.[1/] This practice returns the soil to something resembling its original state in terms of the availability of nutrients and toxic metals to plants. It also increases the sizes and the variety of microbe populations. We have found no evidence that the economics of liming causes farmers to fail to return soils to an approximation of the aforementioned state.

In a verbal communication, N.R. Glass (1979) of the United States Environmental Protection Agency has stated that if all the sulfur dioxide emitted annually east of the Mississippi River were to fall as acid precipitation on the agricultural soils of the region, ". . . a five percent increase in liming would be required." According to a verbal report on January 17, 1980, from Dr. Ed Strobe, Professor of Agronomy at Ohio State University, the 1979 cost of liming, including spreading, in the Ohio Valley region is $6 to $8 per ton. Raising soil pH to 6.0 for most cropping systems (e.g., alfalfa, clover, corn) on average mineral soils requires the application of 2 to 3 tons of lime per acre every 4 to 5 years. There is substantial variation across soil types, however, as Buchman and Brady (1960, p. 419) show. In the 1970's according to the U.S. Department of Agriculture (various issues), the annual consumption of pulverized lime in the United States varied from a low of 26.7×10^6 tons in 1972, to a high of 39.8×10^6 tons in 1976. In real terms, its price, independent of the cost for spreading, was consistently around $3 per ton in 1978 dollars, with the 1972 price being $3.14 and the 1976 price being $3.08. It seems unlikely, therefore, that a five percent increase in biological liming requirements would have much of an effect on either the cost of liming or on the farmer's perceptions of the economically optimal amounts of lime to spread.

Table 1 gives the average acreages, yields, and values of eight major field crops for 1975-77. In addition to Minnesota and the states east of the Mississippi River, Iowa and Missouri are included. With the exception of potatoes, the eight crops listed are the major field crops produced in the

TABLE 2.1

Average Acreage, Production and Gross Value for Selected Commodities, by Region and Total, 1975-77 Crop Year.

USDA Production Regions

	A alachian 1/			Delta 2/			Corn Belt 3/			Northeast 4/			Southeast 5/			Total	
	Acres	Prod.	Value	Acres	Prod.	Value	Acres	Prod.	Value	Acres	Prod.	Value	Acres	Prod.	Value	Acres	Total
								(Millions)									
heat	1.09	35.50	95.50	0.72	25.20	63.20	6.41	254.80	713.70	0.71	24.80	17.11	0.34	9.50	26.90	9.26	973.40
rn	4.35	284.50	715.50	0.26	12.30	27.20	36.10	3513.80	7740.30	2.59	218.90	500.00	3.30	172.80	390.70	46.60	9373.70
arley	0.20	8.60	12.75				0.04	1.62	2.38	2.62	12.50	19.41	0.04	1.01	1.44	2.90	36.00
orghum	0.14	7.12	14.70	0.31	14.80	29.30	0.79	50.20	93.50	-			0.08	2.93	6.34	1.32	143.90
otton	0.40	0.31	78.30	2.60	2.60	696.80	0.24	0.20	51.50	-			0.73	0.58	160,70	4.00	987.30
ugar Beets							0.03	0.62	15.50	0.01	0.02	0.34				0.03	15.80
obacco	0.80	1623.30	1792.10	6/	0.09	0.11	0.02	46.10	50.90	0.04	366.65	52.28	0.16	324.00	356.90	1.01	2252.30
oybeans	4.83	116.70	683.60	10.20	230.40	1370.80	11.84	113.90	4978.70	0.74	18.80	107.20	3.10	87.80	508.20	30.20	“7648.40
lfalfa	0.47	1.22	67.50	0.09	0.23	12.80	4.02	12.50	692.50	2.08	5.46	305.70				6.66	78.50
otal	12.30	7/	3460.00	14.20	7/	2200.30	59.50	7/	14339.00	8.80	7/	1002.10	7.80	7/	1451.20	102.60	22452.60

Source: USDA Agricultural Statistics, 1978. Washington, D.C. 1979

1/ Includes states of Kentucky, North Carolina, Tennessee, Virginia and West Virginia.

2/ " " " Arkansas, Mississippi and Louisiana.

3/ " " " Illinois, Indiana, Iowa, Missouri and Ohio.

4/ " " " Delaware, Maine, Maryland, Massachusetts, New Hampshire, New Jersey, New York, Pennsylvania , Rhode Island and Vermont.

5/ " " " Alabama, Florida, Georgia and South Carolina.

6/ Less than one thousand acres.

7/ Different units of measurement for production preclude aggregation. Specifically, wheat, corn, barley, sorghum and soybeans are measured in bushels, tobacco in pounds, sugar beets and alfalfa in tons and cotton in bales (500 pounds).

indicated subregions. The range of tolerances to acidified soils for these crops varies from highly sensitive (alfalfa) to moderately tolerant (corn). Potatoes are not included because of their relatively high tolerance for acidified soils (pH = 5.0 - 5.4, as previously noted). Table 1 indicates the substantial magnitude of agricultural activity in these five subregions, individually and in the aggregate. The eight field crops in the subregions account for 38 percent of the total harvested acreage of nearly all crops in the United States, with the Corn Belt subregion comprising the single largest area of crop land. Furthermore, these eight crops assume a dominant role in United States agricultural exports. In terms of gross "on-farm" value, the $\$24 \times 10^9$ in receipts represents 25 percent of the total value of all agricultural commodities, including livestock, produced in the United States. In terms of individual crops, corn represents the single largest component of gross value, followed by soybeans.

Assuming that the 117.35×10^6 acres of the field crops in Table 1 would annually require the application of 0.40 to 0.75 tons per acre of pulverized lime in the absence of acid precipitation, anywhere from 46.94×10^6 to 88.01×10^6 tons would be used each year. The lower bound of this interval exceeds the maximum amount (39.83×10^6 tons) ever used for agricultural purposes in the entire United States. We, therefore, assume that annual use in the region of interest in the absence of acid precipitation would be 35×10^6 tons. Thus, if acid precipitation were to add 5 percent to the quantities of lime farmers in the region choose to use, an additional 1.7×10^6 tons would be applied. At \$6 to \$8 per ton in 1978, the annual cost of purchasing and applying the lime would be $\$10.50 \times 10^6$ to $\$14.00 \times 10^6$. This estimate, which is probably exaggerated for several obvious reasons, should, however, be contrasted with the estimate of the Commission on Natural Resources of the National Academy of Sciences (197 , p. 178). The latter employed a 197 cost, including spreading, of \$14 to \$18 per ton for an additional 12×10^6 tons of lime to counter only the effects of acidifying atmospheric depositions. Implicitly, this presumes that from farmers' decisionmaking perspectives, one-third as much lime is required to counter the soil-acidifying effects of these depositions as is required to counter the soil-acidifying effects of inorganic fertilizers and reasonably pristine precipitation.

In addition to its soil acidifying effects, acid precipitation can directly harm plants by causing foliar necrosis, reduction of leaf area, leaching of leaf surface minerals, and cuticular erosion as the plant foliage intercepts the precipitation [Cowling (1978, pp. 49-50)]. The seriousness of these effects in terms of yields is thought to differ widely across plant species and across different life stages of the same plant. However, unless one is willing to make rather tenuous analogies with the well-known effects of sulfur oxides [Committee on Sulfur Oxides (1978, pp. 80-129)], there appears to be only minimal knowledge about the effects of acid precipitation upon crop yields. The two most commonly cited studies referring specifically to acid rain effects on yields seem to be Ferenbaugh's (1976) research on pinto beans, and a degradation in the exterior appearance of Yellow Delicious apples that Cowling (1978, p.59) mentions. These results are hardly a sufficient basis for any analysis, sophisticated or otherwise, of the economic impact of the direct effects of acid precipitation upon agricultural yields. Dose response functions on a greater variety of crops are expected to be available soon, however. The United States-Canada Research Consultation Group on the Long-Range Transport of Air Pollutants (undated, p. 19) reports that results from studies of the sensitivity of field crops to simulated acid precipitation were to be made public in the spring of 1980. In the meantime, the Consultation Group (p. 19) states that "... there is every indication that acid rainfall is deleterious to crops," and that there is "... the potential for widespread economic damage to a number of field crops."

In spite of the current absence of dose-response data to do either an unsophisticated or a sophisticated economic analysis of the direct effects of acid precipitation upon crop yields, one might hazard a guess about the magnitude of the "potential" for widespread economic damage by drawing analogies with other cropping systems that are known to have been exposed to continuing high levels of air pollution. If the forms of representative plant responses and representative farmers' decisions based on these responses are roughly similar across types of air pollutants, combinations of crop types, and geographical areas, the hazarded guess would have some basis in reality.

Adams, et al. (1979) have studied the economic effects of photochemical oxidants in southern California upon twelve vegetable and two field crops. Their study took into account differences in the tolerances of the yields of the various included crops to oxidant exposures, changes in cropping patterns, input substitutions, and locational changes. For many of the included vegetable crops, southern California has a seasonal near-monopoly. Major adjustments in cropping patterns and cropping locations were predicted and observed within the region in response to increasing ambient oxidant levels. After all these adjustments, a 3.01 percent decline in the sum of producer rents and consumer surpluses occurred, with three-quarters of this percentage decline being producer rents. The total on-farm value figure of $\$24 \times 10^9$ in Table 1 for eight field crops in five agricultural subregions of the eastern United States might or might not be a reasonable approximation of the sum of producer rents and consumer surpluses obtained from these crops.[2/] If the 3.01 percent reduction is applied to the $\$24 \times 10^9$ gross on-farm. value, a loss of $\$720 \times 10^6$ results. In 1978 rather than 1976 dollars, this would be about $\$828 \times 10^6$, a figure that is to be taken no more seriously than the credence one is willing to give the analogies and assumptions from which the figure is derived. If one were to include various fruit crops such as apples, oranges, and peaches, legumes and tubers such as peanuts and potatoes, and ornamental in the calculations (in 1977, the U.S. on-farm value of the production of these crops was more than $\$4.0 \times 10^9$), the figure might reach $\$1.0 \times 10^9$.

Forestry Effects

Of all the potential effects of acid precipitation, the effects upon forest ecosystems "seem to be least understood. General qualitative descriptions of what might happen abound, however, e.g., Abrahamsen and Dollard (1978), Cowling (1978), Dochinger and Seliga (1976), and Tamm (1976). As with agricultural systems, the health of forest ecosystems can be affected directly and indirectly by acid precipitation. Moreover, the health states can be aided as well as hindered [Tveite (1980)].

Detrimental direct effects upon the physiological and metabolic processes of forests are likely to occur when foliage intercepts acid precipitation. Reductions in leaf areas, excessive leaching of organic materials, cuticular erosion, necrosis, and reductions in photosynthesis and the cycling of nutrients to other system components are all regarded as likely events. However, Abrahamsen and Skeffington (1979, p.D.2.1) note that carefully documented field cases relating necrosis to acid precipitation do not exist. Our search of the literature has not turned up field demonstrations of the other effects, although thev all have been found for one or more tree species in controlled experimental settings.[3/] All of these studies of particular types of effects fail to make clear how the observed effect is related to tree growth. There nevertheless appears to be a consensus that the physiologically most active developing tissues are the most sensitive [Knabe (1976)1.

Neither the field nor the experimental studies of the effects of acid precipitation on tree growth have yielded consistent findings. In fact, findings of no effects or positive effects of acid precipitation upon tree growth dominate the literature. The results reported in a recent paper by Lee and Weber (1979) are typical. These authors subjected the seeds of eleven tree species, including Douglas fir, eastern white pine, yellow birch, sugar maple, sumac and hickory, to an artificial rain containing enough dilute H_2SO_4 to lower rainfall pH levels to 4.0, 3.5, and 3.0. The plants were exposed to these rains for three hours for each of three days a week over 1.5 years. In general, acid precipitation had either no effect or a positive effect upon the proportion of seeds that germinated and upon the top dry weight and root dry weight of the germinated seeds. The soil solutions in which the seeds were placed had high buffering capacities, causing the authors to infer that whatever effects were observed were direct. In particular, they did find some negative effects upon foliage but they also frequently found enhanced rates of growth. The latter they attributed to fertilization of the soil solution by sulfur, nutrients leached from the plant surfaces, and increased plant uptake of soil nutrients. For Tee and Weber (1979), the "stimulator" effects of acid precipitation over the 3.0 - 4.0 pH interval nearly always dominated or negated the "inhibitory" effects. The results of these authors thus support Abrahamsen's and Dollard's (1979, p.8) statement that: "Statistically significant effects [in laboratory settings] have been observed only when applying 'rain' with pH 3 or lower."

Elsewhere, Abrahamsen and Skeffington (1979, p.D.2.4) indicate that they and their discussants "... could think of no evidence to indicate yield reductions under treatment with acid rain in laboratory conditions at realistic levels of acidity (i.e., pH≅4). Later on, they state that "...no artifical acidification experiment has produced a growth reduction at pHs equivalent to those commonly observed in rain, and some have produced a stimulation." These stimulator effects of acid precipitation (or acidifying deposition) have recently been broadcast in several widely read periodicals, e.g., Maugh (1979). The implicit position would then seem to be that some limited amount of acid precipitation in excess of that which is natural has a positive impact upon plant growth, probably as an amendment to sulfur and nitrogen deficient soils. Given that the time interval over which these positive amendment effects would occur is typically unspecified, one is left to presume they would continue at least as long as the amendments continue.

Forest ecosystem acidification leads to reduced nutrient cycling rates. Not only can nutrients become complexed at low pH levels, as for phosphates, but decomposition of organic material is slowed. Many potential decomposes are less active or inactive at pH levels much below 5.0, and protozoa and earthworms are very rare in soils having pH levels below about 4.0 (Abrahamsen et al. 1976). Lohm (1980) observed that acidification decreased decomposition rates for both needles and litter as well. as decreasing fungal lengths, bacterial numbers and cell sizes. The results of Francis et al. (1980) also suggests that acidification of forest soils may cause significant reductions in leaf litter decomposition and reduce nutrient recycling rates in forest ecosystems by slowing ammonification, vitrification and denitrification. Tamm et al. (1976) found that even moderate additions of sulfuric acid to soils produce obvious effects on nitrogen turnover rates.

Besides affecting nutrients through reduced decomposition rates, depressed environmental pH levels depress nitrogen fixation. Denisen et al. (1976) found that lowered pH levels caused both the nitrogen fixing bacteria Azotobacter and nitrogen fixing blue-green algae to disappear from the soil.

The net consequence for terrestrial ecosystems of decreased nutrient cycling and increased nutrient leaching caused by acidification is reduced productivity [Abrahamsen et al., (1976); Glass and Loucks (1980); Leivestad et al., (1976)]. Such responses have been termed "self-accelerating oligotrophication" by Grahn et al. (1974). Acidification could reduce or remove nutrient pools, constraining the redevelopment of previous biomass levels [Glass, (1978)].

Using nitrogen as his example nutrient, Tamm (1976, p. 237) presents a flow diagram which places all of the preceding results in a most meaningful perspective. We reproduce with minor adaptations his diagram as Figure 1. Tamm (1976) states that the diagram is meant to be a hypothetical representation of the effects of increased strong acid in a system where much of the nitrogen supply comes "... from decomposition in an organic A_0 horizon with a high carbon/nitrogen ratio (>15)." This suggested structure is consistent with experimental. observations in which simulated acid precipitation contributes positively to growth, even though it is expected that, in the long-term, growth would decline once the buffering capacity of the soil is exhausted. Tamm (1976, p. 338) also notes that the structure could account for the frequently observed experimental failure of lime to enhance growth rates: by immobilizing the nitrogen in organic matter, the availability of nitrogen to the trees is reduced.

The only available estimate of the long-term effects of acidifying deposition upon pH levels and base saturation appears to be McFee, et al. (1976). These authors estimated that for a "typical" midwestern soil, precipitation with a strong and disassociated acidity of pH = 4.0 at 100cm annually for 100 years would reduce soil base saturation by 19 percent and lower soil pH by 0.6; at pH 3.7 (a doubling of acidity) and PH 3.0, 50 and 10 years would be respectively required to bring about the same changes. If the soil initially has a fairly high pH level, effects of this magnitude would probably not be noticeable in terms of plant growth results; however, if the soil already has low pH, it is generally thought [McFee (1978, p. 66)] that leaching would rapidly increase. Generally, therefore, it is thought that $d^2(\text{Leaching})/d(\text{pH})^2 > 0$, implying that the rate of loss of soil nutrients would get progressively worse as time passes. It thus seems that insofar as forest soils are concerned, acid precipitation has all the attributes of acquiring possible short-term gains in forest growth at the cost of probable long-term losses in forest soil fertility. For both biological and economic reasons, it seems unlikely that liming can counter the fertility decline. In addition to the previously mentioned biogeochemical argument of Tamm (1976), Tisdale and Nelson (1976, p. 428) note that "... particulate of limestone cannot move in the soil, and consequently they must be placed where they are needed." Tilling lime into extensive areas of forest soils would seem both a technical and an economic impossibility.[4/] In anything other than geological time, accelerated soil acidification, therefore, appears irreversible.

Unfortunately, it is quantitatively unclear how the above reductions in soil fertility will ultimately affect forest yields or the properties of other forest ecosystem components (water storage, game animal populations, aesthetics, etc.) that humans directly value. We were unable to discover any information whatsoever that related acid precipitation to the latter components.[57] Jonsson and Sundberg (1972), in a frequently cited paper, suggest 2 to 7 percent declines due to acid precipitation in the annual growth rates of forests in southern Sweden during the 1950-1965 period relative to 1896-1949. However, Cogbill, .(.1976), in a simple time trend study of comparative tree ring growth in areas of eastern North America subject to and free of acid precipitation, could find no differences attributable to acid precipitation. The trends he observed started no later than 1890, and continued until 1.972-73. However, the credibility of Cogbill's (1976) results must be tempered by questions about the comparability of sites at which precipitation pH has been measured [Perhac (1979)] ; the errors inherent in much of the field instrumentation traditionally employed to measure pH [Galloway, et al. (1979)]; and one study [Frinks and Voight (1976)] which gives cause to believe that, at least in one area of Connecticut, the pH of precipitation has been rather low and unchanged since the early 1900's.

Given the empirical. confusion that exists with respect to the ultimate impacts of acidifying deposition on rates of forest growth, we choose to adopt Jonsson's (1976, p. 842) position that there is "... no good reason for attributing the reduction in growth to any cause other than acidification." We adopt this position because it is consistent with existing knowledge of the biogeochemistry of forest ecosystems [e.g., Likens, et al. (1977)] as well as the economic law of variable proportions.

One's willingness to accept estimates, made using current price and production data, of substantial positive benefits from reducing forest exposures to acid precipitation or acidifying deposition must be tempered by evidence that the current elasticity of substitution between land and intensive forestry is very high. Clawson (1976), for example, argues that the following outputs of the national forest system can all be economically and simultaneously increased as follows: net annual growth can be twice as great; designated wilderness areas can be four times greater; outdoor recreation can be doubled; and wildlife stocks and water storage can be modestly increased. Miller (1978), while considering endangered animal species, also emphasizes that improved management techniques have great potential for maintaining existing wildlife stocks now suffering from environmental stresses. Hair, et al. (1980, pp. 514-519) note that if all opportunities offering at least a four percent net annual return on all forest land. in the South were to be exploited, the region's 1976 net annual timber growth would have increased by 86 percent. In the Northeast and North Central regions, the corresponding increase offering a similar minimum return would have been 25 percent. Berk's (1979) recent finding that private forest owners act as if they faced a real before-tax interest rate of only five percent is consistent with management behavior which fails to exploit the opportunities that Hair, et al. (1980)

think to exist.

In Minnesota and the states east of the Mississippi River, there were, according the the U.S. Forest Service (1978, pp. 1, 97), 229.8 x 10^6 acres of commercial and productive reserved forest land that had a net annual 1977 growth of 6.12 x 10^9 cubic feet of softwoods and 7.51 x 10^9 cubic feet of hardwoods. Lerner (1978, p. 735) indicates that 1977 stumpage prices in 1978 dollars for the softwoods averaged about $1.30 per cubic foot, while for hardwoods they were about 50 cents per cubic foot. Jonnsson and Sundberg (1972) and Panel on Nitrates (1978, p. 577) judge that a reduction of five percent in net annual growth falls within the interval of reduced growth one might reasonably expect from the acid precipitation now falling upon Scandinavia and eastern North America. If the precipitation in Minnesota and the states east of the Mississippi River were pristine ($pH \cong 5.65$), a five percent increase in 1977 net annual growth of softwoods would have amounted to 310 x 10^6 cubic feet. Hardwood net annual growth would have increased by 375 x 10^3 cubic feet. Assuming these increases would not have appreciably affected stumpage prices, the 1978 market value of the additional softwood would have been $404 x 10^6. The hardwood increase would have had a 1978 market value of $188 x 10^6. In 1978 dollars, the 1977 market value of the increase for both softwoods and hardwoods would, therefore, have been $592 x 10^6.

The valued outputs of lands devoted to forests are by no means limited to timber. These lands provide outdoor recreation, aesthetic satisfaction, water storage, wildlife habitats, and a variety of other services. Unfortunately, representative estimates of the value of the sum of these services are rare. The sole immediately and easily useful estimate we have been able to find is the study of Calish, et al. (1978). These authors, in a rather nonrigorous but nevertheless extremely clever and interesting paper, estimated that the 1978 annual non-timber value (in terms of harvestable game animals and fish, water flow, nongame wildlife diversity, visual aesthetics, and prevention of mass soil movement) of a representative Douglas fir forest in the Pacific Northwest was $87 per acre. Assume that this same value per acre can be applied to eastern forests; and further assume that current levels of acid precipitation reduce these non-timber values in the same proportions as was assumed for the timber values, i.e. , by about five percent. Calish, et al. (1978) attribute about 11 percent of the $87 per acre to the production of harvestable fish, implying that $77 per acre consists of non-timber values for which we have not otherwise accounted. Five percent of this $77 is $3.85. Thus, if pristine precipitation were to replace acid precipitation in Minnesota and east of the Mississippi River, these procedures imply that 1978 non-timber values in this area would have annually increased by ($3.85 per acre) (299.8 x 10^6) = $1.15 x 10^9. Adding the timber and non-timber value increases yields an annual benefit of $1.75 x 10^9 in 1978 dollars. Assuming a 15 percent discount rate and that this increase could be sustained indefinitely, a discounted value of $\$11.66 \times 10^9$ results.

Aquatic Ecosystem Effects

Contrary to the forest and forest soil effects of acidification, a substantial amount of economically useful quantitative information is available about the aquatic ecosystem effects of acidification. McFee (1976) conjectures that some soils in Scandinavia and eastern North America have been subjected to acid precipitation for approximately a century. Only within the last decade or two, however, are the soils thought to have become sufficiently acidified to influence the pH levels of inflows to fresh water bodies. Because of this relative immediacy, changes in the biota of these water bodies have been observed rather than being considered as historically preordained. In addition, there has for several decades been an historical record of the impact of acid mine drainage upon the biota in the streams of the Appalachian region. Barton, (1978, p. 314) states that in the United States 10,000 miles of streams and 29,000 surface acres of impoundments and reservoirs ".. . are seriously affected by mine drainage."

Declines to about 6.0 in the pH levels of fresh water bodies reduce primary production. Since primary production is reduced, detritus derived from the decay of plankton tends to disappear. As a consequence, water transparency increases, detrital material decreases, and there are increases in soluble alumina, irons, magnesia, and trace metals such as cadmium and mercury. The phytoplankton and zooplankton species which survive are obviously acid-tolerant, perhaps because they are resistant to heavy metals. However, they also concentrate these metals which, in turn, may make them toxic to many fish and bottom-dwelling (benthic) organisms. Although fish and insect kills occurring during heavy rains and spring snowmelts have been frequently observed [e.g., Gjessing, et al. (1976, p. 65)1, the major effects upon fish and insects are thought to stem from reproductive and recruitment failures caused by calcium metabolism difficulties, the accumulation of heavy metals in parents, and the exposure of those young that are produced to these metals [Fromm (1980)]. Schofield (1976, p. 229) indicates that increased salinity tends to make fish more acid-tolerant, apparently, according to Packer and Dunson (1970), because it enables fish to replace the body sodium losses that low pH causes. The reproduction failures can result in extinction of the species in acidified water bodies. Almer, et al. (1978, pp. 303-307) cite several cases where surviving individuals of some game fish species grew more rapidly and were of larger size than were similar individuals in less acidified water bodies. They attributed this to the lessened competition for the available food stock.

Both Almer, et al. (1978, pp. 308-309) and Gorham (1978, pp. 41-42) remark upon the possible implications of altered (generally reduced) diversity and biological productivity of aquatic flora and fauna for organisms such as amphibians, birds, and mammals who spend important parts of their life cycles in and around aquatic environments or who are dependent upon these environments for some or all of their food supplies. Almer, et al. (1978), tell of fish- eating birds such as loons who have migrated from acidified lakes in Sweden to sites with more ample food supplies. Gorham (1978) expresses concern for the impact upon moose populations and distributions if production of the aquatic plants forming parts of their diets is inhibited. Birds of prey, such as eagles and osprey, and game birds, such as mallards and wood ducks, could be subjected to altered populations and distributions from reductions in their aquatic food sources. Little seems to be known, however, about the ease with which these animals could substitute nonaquatic food sources.

In the following pages several attempts will be made to develop rough economic values for the aforementioned aquatic ecosystem effects. Each attempt is made in order to exploit a particuar type of natural and/or economic information. The measures developed are not additive. Most important, as is the case throughout this report, all the measures are and very sensitive to minor perturbations in assumptions.

Although the criteria he uses for determining what is and is not "fish-able" are unclear, Todd (1970, p. 303) shows 19.14×10^6 acres of fishable fresh-water streams and lakes in Minnesota and the states east of the Missis-sippi River. This excludes the Great Lakes which have an area of 38.00×10^6 acres. Adams, et al. (1973, p. 43) indicate that about 23 percent of the approximately 168×10^6 people twelve years or older then living in this area passed an average of 7 days fishing in the summer quarter of 1972. This represents 270×10^6 fishing activity days. No reliable information could be formed giving the seasonal distribution or the fresh-salt water distribution of fishing activity days. We, therefore, assume that 85 percent of this fresh-water fishing occurs during the summer quarter. Upon making the adjust-ments called for by these assumptions, we are left with 287×10^6 fishing activity days in Minnesota and the states east of the Mississippi in 1972. We assume that the number of activity days in 1978 was 300×10^6 more-or- less.

A number of studies of varying degrees of sophistication are available which purport to give the representative willingness-to-pay for an additional fresh-water fishing activity day. These estimates are distributed over a range from 10 to 30 mid-1970's dollars. The S20.72 uncompensated consumer surplus estimate obtained in 1972 by Gordon, et al. (1973) is adopted here. Assuming that increases in real income, in the value of time, and changes in relative fishing costs have not altered this figure, this amounts to $32.30 in 1978 dollars. If the marginal value of a fishing activity day is a constant regardless of the availability of opportunities to catch fish and if no fishing occurs in the absence of fish, the extinction of fresh-water fish life in Minnesota and the states east of the Mississippi River would have resulted in economic losses of (300 x 10^6 activity days) ($32.30) = $9.69 x 10^9 in 1978. Assuming the number of fishing days is linearly and inversely related to the acres of fresh-water having fish populations, and if Tables 1 and 2 from Chapter III are to be believed, well over half this loss would occur before all fresh-water in the aforementioned area reached a uniform pH level of 6.0. If large regions in the general area retained pH levels for fresh- water at 6.5 - 8.0, even though the average pH over the entire region was 6.0 or less, the economic losses would be a great deal less. This is because fishermen would readily be able to substitute away from an acidified to nonacidified bodies of water.

The recreational value of the fresh-water resource in Minnesota and the states east of the Mississippi is clearly rot limited to fishing activities. One major additional use is for hunting, particularly waterfowl hunting. According to Todd, (1970, p. 303), there exist in this area 48.83 x 10^6 acres of natural wetlands "... of significant value to fish. and wildlife." In 1980, the U.S. Water Resources Council (1968) projected that the number of water-related hunting activity days in the area would be 220 x 10^6. This was projected from the 162 x 10^6 days taking place in the same area in 1960. On the other hand, the U.S. Fish and Wildlife Service (1972, p. 31) estimated the number of 1970 waterfowl hunting activity days by people 12 or more years old to be 17.58 x 10^6. Since, relative to fishing activity days, this seems more plausible, we employ it here. Hammack and Brown (1974, p. 29), in their study of the value of waterfowl, found that the average waterfowl hunter passed 9.7 days each season engaged in the activity, and acquired a consumer surplus of $247 (= $462 in 1978 dollars) from the right to hunt during the 1968 season.

Even if fresh-water pH levels were universally to drop to 4.0 or less throughout the region east of the Mississippi, it cannot be assumed that all waterfowl in the region would disappear. Some invertebrates that constitute part of the diet of waterfowl would survive. More important perhaps, there are land-based food items (such as agricultural grains) that waterfowl might employ as a substitute food source. Nevertheless, it should be noted that several very important hunted species of waterfowl currently get 50 percent or more of their food supplies from aquatic sources. Martin, et al. (1961) state, for example, that 50 to 70 percent of the diets of the mallard, the black, and the common goldeneye ducks consist of aquatic insects, crustaceans, and mollusks. Moreover, the habitats of these birds, especially the mallard, tend to be the ponds and shallow lakes thought to be more susceptible to acid precipitation. Of the waterfowl hunters in the Hammack and Brown (1974, p. 39) sample, 47 percent stated that mallards were their first preference in waterfowl hunting.

Given the almost total lack of information on the impact of fresh-water acidification upon the population and distributions of waterfowl, we choose to make some quite arbitrary assumptions. In particular, we assume that a representative member of a representative species currently obtains half its food supply from fresh-water aquatic environments. It, therefore, has some ability to substitute food from terrestrial or marine environments for the fresh-water source without having to alter its location in any given season. Without guidance from any source, we assume that the destruction by acidification of the acid-intolerant portion of the fresh-water food supply for waterfowl will result in a reduction of 20 percent in the waterfowl population during the hunting season in Minnesota and the states east of the Mississippi River. Further assume that the elasticity of waterfowl hunting activity days with respect to waterfowl populations is 0.5. If the number of waterfowl hunting activity days in the region of interest was approximately 30×10^6 in 1978, the assumed elasticity and reduction in waterfowl population implies a drop in 1978 activity days to 28.5×10^6. If the marginal value of an activity day is a constant, the Hammack and Brown (1974) estimate of the consumer surplus obtained from the right to hunt waterfowl implies an activity day valuation of $462.00/9.7 = $47.63. The ten million day reduction in waterfowl hunting activity days, therefore, yields a 1978 economic loss of $\$71.45 \times 10^6$. Given our long chain of arbitrary assumptions, we conclude that an estimate of 70 million dollars is reasonable. It is important to note, however, that Hammack and Brown (1974, p. 63) found that waterfowl hunters could annually shoot 76 percent of the waterfowl in the Pacific flyway without reducing the breeding population. It seems unlikely that fresh-water acidification would have an effect upon recruitment of this magnitude. One might, therefore, conclude that fresh-water acidification will cause no changes in waterfowl populations and distributions and thus no economic losses will be incurred. In cases where waterfowl and fish have been competing for the same aquatic insects, it may be that acidification by reducing fish populations will reduce competition for the aquatic insect food source and thereby enhance waterfowl populations.

If the food supplies of waterfowl are harmed. by fresh-water acidification, it follows that the food supplies of birds that are not legally hunted will also be harmed. The diet of the osprey consists entirely of fish, while bald eagles are mostly dependent upon fish for food. Cranes, including the sandhill crane, utilize a variety of aquatic vertebrates and invertebrates for food sources. In spite of the possibility that these and other birds will suffer population declines from fresh-water acidification, we have no basis whatsoever on which to formulate a guess at the economic losses a population decline might entail. Not only do we lack any quantitative information on population responses to acidification, we also lack any information about the values people place upon encounters with these birds.

A somewhat similar problem exists for mammals dependent upon aquatic ecosystems for part of their food or even for their everyday habitats. These would include, for example, racoon, mink, muskrat, marten, beaver, river otter, and even moose and black bear. No readily available information exists on the population responses of these animals to acidification-induced perturbations in their aquatic food sources or habitat. Nor does any economic information exist on the value of an encounter with these animals. Economic information does exist, however, on their market value when they are killed for their furs. The U.S. Water Resources Council reports according to Todd (1970, p. 270) that in 1966, 7.0×10^6 "fresh-water dependent" fur-bearing animals were captured in Minnesota and the states east of the Mississippi. These animals, in 1978 dollars, had a market value of $\$26.7 \times 10^6$, or approximately $3.80 per animal, assuming the resources employed in their capture had no valuable alternative uses. Measured solely then in terms of the worth of their fur when captured, the economic impact of a fresh-water acidification- induced population reduction will be minor. Again, however, this disregards the animals' value in terms of simple observations during encounters as well as the value they contribute to hunting recreational activities.

Boating and swimming are additional recreational activities that conceivably could be affected by reductions in the pH of fresh-water. However, in a study of the acid mine drainage problem in the Appalachian region, Robert R. Nathan Associates (1969) was unable to find any variation in boating and/or swimming activity days with respect to different levels of pH. This is not too surprising since it is generally recognized that boaters and swimmers respond negatively to increased turbidity. Kramer (1978, p. 354) notes that there is an approximate doubling in transparency, measured as Secchi depth, for each unit decrease in pH over the 6.5 - 4.5 pH interval. He attributes this "... to the dissolution of iron and manganese colloids and the decrease in organic detritus with decreasing pH due to decreasing photosynthesis" (p. 354).

Added to all the above results must be the commercial value of captured fresh-water fish. In 1965, Todd (1979, p. 270), using data from the U.S. Water Resources Council, estimated that 2801.6×10^6 pounds of "fresh-water-dependent" fish were caught in our region of interest. The market value of these fish was $\$196.2 \times 10^6$ (=$\$404.2 \times 10^6$ in 1978 dollars). Sixteen percent of this catch, was either fresh-water or anadromous species. Assuming the resources employed to catch this sixteen percent had no valuable alternative uses, if these fresh-water or andromous species were to be extinguished, the annual loss would amount to 65×10^6 1978 dollars. This includes the extinction of the catch from the Great Lakes.

When all the preceding effects of acid precipitation upon fresh-water ecosystems in Minnesota and the states east of the Mississippi River are summed, one obtains 1978 annual benefits for preventing pH levels falling below 4.5 - 5.0 for all fishable fresh-water bodies in this area of $10 to $11 billion. Nearly all the benefits are attributed to the maintenance of recreational fishing opportunities. It must nevertheless be emphasized that the economic. and biological analyses on which these estimates are based have weak. analytical foundations. Given those limits, somewhat more creditability might be achieved by adopting a different approach to assessing the benefits of acid precipitation control.

If the acidification of fresh-water does negatively influence ecosystem attributes that human beings value, one would expect these negative influences to be capitalized into land that offers ready access to these valued attributes. In particular, economic theory predicts that land prices will be consistent with the values people attach to the differences in these advantages of access. Freeman (1979) outlines the circumstances in which these prices will reflect the variations in consumer surplus generated by the differences in access advantages.

Adams, et al. (1973, p. 111-110) present estimates by farming region of differences in the 1972 per-acre value of recreation land with and without water. For the states east of the Mississippi River, the median difference by region appears to be about $1,250 per acre. Recreation land without water is generally about 33 to 50 percent the value of land with water. If the values of both types of recreational land behaved as did farm real estate values between 1972 and 1978, their values approximately doubled [Economic Research Service (1978)], implying that the $1,250 per acre difference in 1972 had increased to a $2,500 per acre difference by 1978.

Excluding the Great Lakes, Todd (1-970, p. 301) finds that there exist 21.86 x 10^6 acres of inland water available for recreation in Minnesota and the states east of the Mississippi. This is somewhat greater than the 19.14 x 10^6 acres that he considers to be "fishable" (p. 303). Of this "fishable" acreage, 3.83 x 10^6 is in streams, leaving 15.31 x 10^6 acres in natural and man-made lakes, exclusive of the Great Lakes. Assume that each surface acre of fishable freshwater lakes, excepting the Great Lakes, is associated with one riparian acre. This one acre figure is a judgment formed by using a table in Todd (1970, pp. 126-127) of the surface acreages and shoreline lengths of the largest lakes in each state of the United States. Man-made lakes tend to have shoreline mileages about 10 times as great as the square mileages of their surface areas, while the shoreline mileages of natural lakes tend to be about half as great as the square mileages of their surface areas. We estimate then, that of the nearly one billion acres of the surface of the globe contained within the boundaries of Minnesota and the states east of the Mississippi, 15.31 x 10^6 (or 1.6 percent) of them are riparian to lakes and ponds.

Added to the lake and pond riparian acreage must be the acreage that is riparian to rivers and streams. Minnesota and the states east of the Mississippi contain about 36 percent of the 260,000 stream miles in the 50 U.S. states, meaning that they have about 94,000 stream miles. Assuming 50.66 riparian acres per stream mile, these 94,000 stream miles yield 4.76×10^6 riparian stream acres. By way of contrast, this means that we estimate 1.24 of riparian acres per acre of stream surface water, as opposed to the one riparian acre estimated for each acre of lake and pond surface water. Upon summing the estimated stream and lake and pond riparian acreages for Minnesota and the states east of the Mississippi River, we obtain 16.55×10^6 acres.

Now let us consider under some extremely strong assumptions what the extinction of fish life on these 16.55×10^6 acres might mean for their values. According to Unger, et al. (1976, pp. B.6 - B.8), there were 1.70×10^9 fresh-water related recreation activity days in 1970 in the United States, of which 0.63×10^9 days, or 37 percent, were devoted in some part to fresh-water fishing. Of the estimated $2,500 per acre difference in 1978 between the values of recreational land with and without ready access to water, we presume that exactly 37 percent, or a $925 premium is attributable to the fishing access the former offers. If the game fish in all of the waters to which the above mentioned acreages are riparian were simultaneously to disappear forever, the total value of these acreages would then decline by $(\$925)(16.55 \times 10^6) = \$15.32 \times 10^9$.

Earlier, using recreational data on fishing activity days and the consumer surpluses associated with them, we estimated that the disappearance of all game fish would have generated annual losses of $\$9.69 \times 10^9$ in 1978. If these losses were to continue in perpetuity, and if they were all to be capitalized into the values of riparian acreages, a discount rate in excess of 200 percent would have to be applied in order to yield a present value of only $\$15.32 \times 10^9$.[6] This hardly seems realistic. When a more reasonable discount rate of 15 percent is applied to this presumed present value for riparian property losses of $\$15.32 \times 10^9$, one obtains annual losses of $\$2.0 \times 10^9$. Lower discount rates imply still lower annual losses. For example, a discount rate of 5 percent implies annual losses of only $\$0.73 \times 10^9$. On the other hand, if one applies a 15 percent discount rate to the earlier estimate of $\$9.69 \times 10^9$ in annual losses, one obtains a present value for this stream of losses of $\$74.29 \times 10^9$, a considerable amount of wealth indeed.

Before rejecting the previous $\$9.69 \times 10^9$ annual loss estimate in favor of an annual loss estimate derived from an equally limited treatment of the annual losses implied by reductions in riparian property values, it is important to keep in mind that these latter losses are related only to the losses in consumer surplus suffered by the riparian property owners plus the fees they might collect from present and future fisherman who wish to gain access to fresh-water bodies over present owners' riparian lands. In the great majority of cases, these fees are not collectable because of the costs of policing access. One can thus make a strong argument that the impact of acidification upon riparian property values, even if the structural conditions outlined by Freeman (1979) are fulfilled, must be a substantial underestimate because of the failure to register the surpluses accruing to fisherman who have free access over the land.

The preceding accounting, in addition to its rather severe limitations in terms of economic analysis, fails to consider the actions private and public bodies might undertake to ameliorate the effects of acid precipitation. In particular, one might lime and/or restock acidified fresh-water bodies. The ecological evidence for the likely restorative successes that can be achieved by liming is mixed. As for restocking, if one is willing to devote the requisite resources, it is perhaps a feasible technical alternative. Whatever the technical and financial feasibility of either or both of these restorative procedures, it must be recognized that some set of decisionmaking bodies must be formed and maintained to implement the restorative procedures. Given the common property attributes of that which is to be restored and the public good nature of the restorative and restocking actions, one might reasonably have serious doubts about whether effective massive restoration and restocking programs can be formulated and implemented.

In principle, the liming of fresh-water bodies will simultaneously serve to raise the pH values of the water and to make the heavy metals abundant at low pH substantially less available biologically. Hagerhall (1979) estimated that in 1973 it would cost $\$45\text{-}70 \times 10^{6}$ to acquire and apply one million tons of $CaCO_3$ in Sweden. Assuming similar costs in the United States, this amounts to \$66-103 per ton per year in 1978. He also states that apparently a one-time application of $30\text{-}50 \times 10^{6}$ tons of $CaCO_3$ (p. 10) would suffice to return $22.24\text{-}27 \times 10^{6}$ acres of Swedish lakes to their pH states at the beginning of the 20th century, if acid precipitation were to cease. Simply to counter the current yearly increment in acidification over this lake area would, according to Hagerhall (197,9, p. 10), require the annual application of one million tons of $CaCO_3$. Thus restoration by liming in 1978 to pH levels of the early 20th century would require a one-time outlay from $(30 \times 10^{6}$ tons$/27 \times 10^{6}$ acres$)(\$66) = \$73.33$, to $(50 \times 10^{6}$ tons$/22 \times 10^{6}$ acres$)(\$103) = \$234.09$ per acre. The mid-range of this interval is \$150 per acre. Thus, always remembering the limits of the analysis, if all the 19.14×10^{6} acres of fishable fresh-water bodies in Minnesota and the states east of the Mississippi River were to become acidified in a fashion similar to the aforementioned Swedish lakes, a one-time 1978 outlay for liming of $\$2.87 \times 10^{9}$ would be required to return them to pH levels in excess of 6.0. However, if the number of acidified fresh-water acres is extended to the natural. wetlands ".. . of significant value to fish and wildlife" [Todd (1970, p 3031, this estimate increases to !(48.83×10^6 acres) + (19.14×10^6 acres)] \$150 = \$$10.20 \times 10^9$. On an annualized basis, using a 15 percent rate of discount, these one-time outlays are respectively equivalent to \$$430 \times 10^6$ and \$$1.53 \times 10^9$.

Assuming the \$150 per ton cost figure for the purchase and application of $CaCO_3$ to be reasonable, a rough check on the above estimates can be obtained by exploiting a statement of Holden's (1979, p. 11).

> "An alkalinity equivalent to 2.5 ugl^{-1} calcium carbonate requires the solution of 2.5 g. $metre^{-3}$. A lake of 10 ha with a mean depth of 2 m (a small lake), requires 500 kg of dissolved calcium carbonate, and probably ten times this amount of the solid to obtain sufficient in solution. A stream flowing at 1 $m^3 s^{-1}$ would require about 80 tonnes $year^{-1}$ in solution. These quantities must be maintained each year. .."

Assume that the lake Holden (1979) describes is representative of most lakes in Minnesota and the states east of the Mississippi River. Further, assume that, because of the greater ability of streams to dilute materials, that representative streams annually require the injection of twice as much $CaCO_3$ as do the representative lakes. Given that "sufficient" solution of $CaCO_3$ requires the injection from all sources of ten times as much $CaCO_3$ in solid form, this is stating that each acre of our representative lake annually requires the introduction of about 450 pounds of $CaCO_3$. Streams, therefore, by our assumption, require 900 pounds of $CaCO_3$ for each acre of their surface areas. The 15.31×10^6 acres of fishable freshwater lakes and the 3.83×10^6 acres of fishable fresh-water streams in Minnesota and the states east of the Mississippi River, therefore, require the annual introduction of (15.31×10^6 acres) (450 pounds) + (3.83×10^6 acres) (900 pounds) = 5.17×10^6 tons of $CaCO_3$ or equivalent. If all this $CaCO_3$ had to be introduced by man, and if each ton of CaCO cost \$150 to acquire and inject, the total annual cost in 1978 would be \$$7?76 \times 10^9$. This figure is much greater than even the one-time cost earlier obtained using the data of Hagerhall (1979). Note, however, that the estimate from Holden (1979) assumes that all $CaCO_3$ entering these fresh-water bodies is somehow to be supplied by man. If, for example, half the $CaCO_3$ necessary to raise the pH level of a fresh-water body is naturally supplied from the catchment area, the \$$7.76 \times 10^9$ cost estimate would have to be reduced accordingly. However, if the 48.83 acres of wetlands "... of significant value to fish and wildlife" are taken into account, and if one assumes that the residence time of water in these wetlands is twice as long as in Holden's (1979) representative lake (implying that only half as much $CaCO_3$ is required as in the representative lake), then an additional (48.83×10^6 acres)(225 pounds) = 5.49×10^6 tons at an additional total cost of \$$8.24 \times 10^9$ would be required annually. Thus, at least in terms of the sets of assumptions employed here, the use of Holden's figures on liming requirements leads to an estimate of the annual cost of liming more-or-less similar to the value of the fishery that would be lost if fresh-water acidification were allowed to proceed unhindered.

The literature occasionally mentions two other alternatives to ameliorating the effects of extant pH. For example, Swarts, et al. (1978) and Leivestad, et al. (1976) find differences within fish of the same age and species with respect to their ability to tolerate low pH. This raises hopes for natural selection processes serving as a means to maintain natural populations, given, of course, that the rate of toxic acidification does not outstrip the rate of natural genetic improvement in tolerance.

Alternatively, the awareness of differences in acid tolerance among species raises the prospect, [e.g., Schofield (1976, p. 230)] for the selective breeding or genetic engineering of acid-tolerant individuals. These individuals would then be used to restock acidified waters. However, if acidification also has a major impact upon the populations of other fresh-water organisms, it is unclear what the restocked fish would eat other then their peers. A similar comment applies to all restocking programs, whether or not with acid-tolerant individuals. Almer (1978, p. 307) nevertheless mentions several species in acidified waters which have substituted surviving invertebrates for their usual food which consists of other fish. It thus seems worthwhile to make some estimates, even if exceedingly rough, of the cost of restocking fish populations.

Bennett (1971, p. 89) presents a table showing average standing crops of various species of fish per acre of fresh-water. The species range from trout and channel catfish to suckers and carp. The table was constructed to show the relative masses of the 19 species listed, given that some species are usually seen in combination with other species. The average of the mean pounds per acre for the 19 species is approximately 36. If there are 19.14 x 10^6 acres of fresh-water lakes and streams in Minnesota and the states east of the Mississippi River, this implies that there are 689 x 10^6 pounds of fish in these waters. According to Lerner (1974, Table 332) the Federal government spent \$298 x 10^6 in 1973 for the maintenance of fish and wildlife populations. Part of this money was spent to propogate and distrribut, according to the U.S. Fish and Wildlife Service (1974, p. 27), 303 x 10^6 fish eggs. If, on average, 5 percent of these eggs survive to one pound adults, and if all the above \$298 x 10^6 1973 Federal expenditures were for fish, then each pound of adult fish cost the Federal government \$19.67 (= \$28.91 in 1978 dollars). For obvious reasons, this is an overestimate. Nevertheless, even if the survival rate is increased and/or the cost per adult fish is reduced, a substantial annual outlay remains. Given the extreme crudeness of the calculation, the difference from the \$10-11 x 10^9 estimated annual value of a loss of the entire game fishery in the region of interest does not seem sufficent to conclude, tentatively or otherwise, that the annual value of the impact of acid precipitation upon aquatic ecosystems is the cost of restocking game fish populations.

Health Effects

The Safe Drinking Water Committee (1977, p. 439) defines "hard" water as that containing 75 mg/liter or more of calcium carbonate or the equivalent. Increased hardness is indirectly associated with elevated pH. Although the Committee does not adopt an unequivocal position, it does state that the body of evidence for "soft" water being a causal agent in cardiovascular disease "... is sufficiently compelling so that 'the water story'is plausible . .." (p. 447). Soft water has a relative lack of inorganic solute health-supporting agents such as calcium, magnesium, and manganese, and a relative abundance of health-degrading metal agents such as cadmium, lead, copper, and zinc. In addition, the effectiveness of several standard methods (chlorination, filtration and sorption, etc.) for reducing concentrations of bacteria, viruses, and protozoa in water intended for human internal consumption sometimes varies directly with pH.

Our brief review of the available evidence makes us reluctant to map disease incidence into the pH levels of drinking water. Nevertheless, it is worth noting that quite small incidence due to the inorganic and organic solutes and the microbiological agents whose human health-degrading potentials are activated by low pH levels can result in large economic losses. For example, about half of the two million annual deaths in the United States are attributed to the various cardiovascular diseases. If only one percent of these one million deaths were indirectly attributed to low pH drinking water supplies, a toll of 10,000 deaths would result. In this context, it is worth noting that the Safe Drinking Water Committee (1977, p. 447) states: "On the assumption that water factors are causally implicated, it is estimated that optimal conditioning of drinking water could reduce this annual cardiovascular disease mortality rate by as much as 15% in the United States." Recent economic research [e.g., Thaler and Rosen (1975)] indicates the value of safety from death. to be about \$500,000 - \$1,000,000 in 1978 dollars. Using the lowest point in this range along with the one percent mortality assumption, would then result in annual economic benefits of $\$5 \times 10^9$. This figure would be increased substantially if one were to account for the losses in life- cycle earnings (and implicitly in labor productivity) due to cardiovascular diseases that do not now and perhaps never will result in death. Bartel and Taubman (1978), while working with a panel of 40-50 year old male twins, found that those with cardiovascular diseases had their annual earnings reduced by 20 - 30 percent relative to their healthy peers. According to the National Center for Health Statistics, 15.7 percent of the

1976 U.S. population suffered from cardiovascular diseases [Lerner (1978, p. 120)].

The preceding makes it appear that the health impacts and consequent economic effects of reduced pH in water used for internal human consumption could readily be very considerable. This assumes, however, that no ameliorative measures are available to either the consumer or the supplier of the water. If at a cost less than the value of the health effects, these ameliorative measures are able to raise pH to the levels that would exist in raw water supples in the absence of above-background acid deposition levels, then the proper health effects benefits to ascribe to the control of acidifying deposition are the ameliorative measure costs avoided.

Less than one percent of the water used by a community is consumed internally by humans [Safe Drinking Water Committee (1977, p. 104)]. Assuming that water consumption and fluid consumption are at most trivially different, this amounts to 1,000 - 2,400 ml/day for adults under "normal" conditions, and 1,000 - 1,670 ml/day for children aged 5 - 14 years [Panel on Low Molecular Weight Halogenated Hydrocarbons (1978, p. 164)]. A representative figure for the entire U.S. population might be 1,900 ml/day or 2 U.S. gallons per capita/day. In 1970, 29.42 x 10^9 gallons were withdrawn in the U.S. daily for rural domestic (2.39 x 10^9 gallons) and municipal (27.03 x 10^9 gallons) uses [Economic Research Service (1974, p. 37)]. Of these withdrawals, it is probably safe to assume that all were treated as if they were to be used for internal human consumption. Assuming no differences in per capita withdrawals for rural domestic and municipal uses across the United States, 1970 withdrawals in Minnesota and in the states east of the Mississippi River were 19.12 x 10^9 gallons/day. Now make the strong assumption that all ground and surface waters east of the Mississippi had pH levels (and associated inorganic and organic solutes and microbiological agents) requiring the addition of 21.6 kilograms of lime per million liters (per 264,175 gallons). This figure, which is due to Randall, et al. (1978, p. 66), refers to a thinly populated region in eastern Kentucky whose water supplies suffer from acid mine drainage. It therefore possibly has a low pH problem resembling that which would occur elsewhere if acid deposition became severe and widespread. Moreover, because of its small human population, the area is unable to fully exploit economies-of scale in its water treatment activities. Its unit costs of treatment are thus likely higher than in larger urban areas.

Randall, et al. (1978, p. 66) estimated the average market price of the "lime" used for water treatment in their study region to be $0.095 per kilogram, presumedly in 1976 dollars. In assigning a cost to the liming treatment, they did not attribute any additional costs to the maintenance and operation of treatment facilities since the facilities would be required even if there were no low-pH problems. Assuming constant unit and, therefore, marginal costs, the total cost of providing lime treatment for 264,175 gallons of raw water would be $2.05. Given our previous assumptions about the distribution of the U.S. population and water withdrawals for rural domestic and municipal uses, this implies that the daily cost of lime treatments for raw water supplies east of the Mississippi River would be about $148,000. This amounts to an annual cost of $54 x 10^6 1976 dollars or $62.1 x 10^6 for 1978. Of course, given the existence of acid mine drainage in important watercourses of the region as well as natural acidification due to soil leaching, much of this cost burden would have to be borne independently of any acid precipitation problem. Thus, the external costs (the environmental costs of increased mining activity for $CaCO_3$ and its equivalents and the negative health effects) of increased liming of raw water supplies would have to be extremely large (at least as much as the value of the negative health impacts of increased acidity) to justify a refusal to ameliorate the health impacts of acid deposition by the liming of raw water intended for internal human consumption.

Household, Commercial, and Industrial Water Supply System Effects

Reductions in the pH levels of water supplies may cause corrosion in household, commercial, and industrial water conveyance systems and water-using appliances, thereby shortening their useful lifes and reducing the flow of their services while in use. In the absence of ameliorative measures, the potential economic losses from this corrosion could be severe. On the other hand excessively high pH levels can have similar effects due mainly to mineral deposits forming on the interior surfaces of the systems and appliances.

Several studies are available that assess the impact of increased levels of total-dissolved-solids (TDS) and/or water hardness upon the economic lifetimes of household and commercial water supply and use systems. Using an eight percent discount rate, d'Arge and Eubanks (1976) estimate 1975 economic loses for a typical Los Angeles household to range from $620 to $1,010 in present value terms for an increase in total dissolved solids from 200 to 700 mg/l. This estimate is three to four times higher than estimates developed by Tihansky (1973) for a similar TDS range throughout the United States. In an appendix to their study, d'Arge and Eubanks (1976, pp. 274-275) used data from Black and Veatch (1967) to explore the extent to which the ratio of TDS to total hardness was important to the useful lifetimes of household conveyance systems and water-using appliances. They found that increases in the ratio made a statistically significant positive contribution to the lifetime of garbage grinders and a statistically significant negative contribution to the lifetime of wastewater pipes. Total hardness, when entered as a separate explanatory variable in a multiple regression, was negatively associated with the useful lifetime of faucets but the association was not statistically significant. It thus appears that the acidification of water supplies, to the extent that it "softens" water in areas with "hard" raw water, could have economically beneficial effects upon water conveyance systems and water-using appliances. Nevertheless, if the effects of excessive "softness," as induced by low pH, upon useful life are more-or-less symmetrical to those of excessive hardness, the economic impacts upon households, and commercial establishments could be considerable.. Since the ferro-alloys, copper alloys, and brasses used in household and commercial water supply systems and water-using appliances are also found in industrial systems and equipment, substantial economic impacts could also be expected in these sectors.

In spite of the potentially large economic impacts of low pH water upon household, commercial, and industrial. water supply systems and water-using appliances and equipment, it does not appear useful to try to calculate the magnitude of these impacts. The reason is that inexpensive neutralization techniques using hydrated or calcined limes are readily available.

In the health effects section, we have calculated the cost of liming rural domestic and municipal water supplies. According to Todd (1970, p. 312), the "optimal" pH levels for domestic water supplies are about neutral (pH = 7.0), although (p.320) the median for the 100 largest U.S. cities in 1962 was pH = 7.5. Thus, given that all rural domestic and municipal water supplies are treated as if intended for internal human consumption, acidification of raw water will have no extraordinary effects upon household and commercial conveyance systems and appliances.

Many industries supply and treat their own process water. The USDA Economic Research Service (1974, p.37) estimates that self-supplied industrial water withdrawals (excluding steam-electric power) from fresh surface and ground sources in the U.S. in 1.970 were 45.87×10^9 gal./day. According to Todd (1970, pp. 329330), most industrial processes require or are indifferent to water with pH in the 6.0 to 9.0 range. In 1968 [Todd (1970, p. 221)], about 72 percent or 33.03×10^9 gals./day of these withdrawals occurred in the states east of the Mississippi River and in Minnesota.

Assume that all of the self-supplied industrial fresh water east of the Mississippi and Minnesota has been acidified to the 4.0-4.5 pH range prior to withdrawal. Further assume that in order for it to be used as a process water its pH must be raised to an average 7.5-8.0 across industries, and that it would otherwise require no treatment prior to use. Thus, excluding the possibility of tying into municipal systems, each plant will have to construct and operate its own treatment facility. Our presumed necessary increase in pH happens to correspond to the pH increases experienced with several acid mine water treatment plants. For example, Bituminous Coal Research, Inc. (1971, pp. 133-134) reports total 1970 capital and operating costs of 97.3, 35.3, and 26.5 cents/1,000 gals. with a plant respectively processing 0.1, 1.0, and 7.0

million gallons/day. They also report on another plant which experienced 1970 total capital and operating cost of 13.6 cents/1,000 gallons for treating an average of 4.0 million gallons/day. In each case, the pH of the acid mine water was raised from about 4.5 to more than 7.0. Barton (1978, pp. 351-354) summarizes the experiences of one mine where water were raised from 4.0 or less to more than 7.5. The commercial operation with 12,800 tons of lime, processed 3.4×10^9 gallons of water with an original pH of 4.0 and a finished pH of 7.9 at 20 cents/1,000 gals.

Using a very fine and therefore more costly limestone slurry, the pilot plant raised a mine discharge of 2.8 pH to 7.4 pH at estimated 1970 capital costs of $55,000 to $766,000 for 100×10^3 to 600×10^6 gals/day operating capacities. Estimated 1970 operating costs, including amortization, were respectively 44 cents to 2 cents/1,000 gals. In the words of Barton (1978, p. 352):

> "Limestone could be the preferable choice for treating nearly all but the most severly loaded discharges. It has the advantages of availability, lower cost, reduced hazards, ease of application, simplicity of plant design, impossibility of water overtreatment, ease of storage, and higher solids concentration of the precipitated sludge."

A review of Todd (1970, pp. 246-274) makes it appear that the median water-using industrial establishment withdraws about 1.0 million gals./day. We therefore estimate, on the basis of the material presented in the preceding paragraph, that, including amortization of capital facilities, a representative 1970 total cost of raising 4.0 pH fresh water to 7.5 pH would be 50 cents/1,000 gals./day. On a yearly basis, therefore, the 1970 total annual cost of treating the 33.03×10^9 gals./day of self-supplied industrial water withdrawn east of the Mississippi could be $5.66 $\times 10^9$ (=$9.51 $\times 10^9$ for 1978). Even though it is fair to presume that the unit cost of lime and treatment plants might increase with an increase in demand of the magnitude posited here, it should also be recognized that the posited increase in demand is also probably vastly exaggerated. Not all fresh water east of the Mississippi is likely to suffer a reduction in pH to 4.0 or even 5.0. Many industries are fairly indifferent to quite low PH. For example, in a survey of the impact of acid mine water upon patterns of industrial water use in the Appalachian region, Whitman, et al. (1969) found that the most impacted industry, primary metals, saw fit to raise the pH level of its cooling water only to 5.0. This was accomplished at minor cost by integrating lime treatments with otherwise existing water treatment facilities. Todd (1970, p. 345) makes it appear that possibly half the industries in the United States in 1959 had their own treatment facilities. Finally, we have not considered possible substitutions from fresh to saline water sources. [7/] The stated estimate might therefore readily be exaggerated by more than an order of magnitude. One can be quite sure that it is not biased downward. Given the industrial treatment facilities already in place, we prefer to treat as trivial the additional costs of treatment attributable to acid precipitation.

Materials Effects

Several studies of the economic impact of air pollution upon the useful lifes of materials have been published, e.g., Gillette (1975), Commission on Natural Resources (197 , pp. 616-619, 695-699), Kucera (1976), and Salmon (1972). Nriagu (1978) presents an extremely thorough review of the physical science literature on the effects of sulfur pollution on materials. A brief review of the material effects of nitrate aerosols is available in Panel on Nitrates (1978, pp. 417-418).

Acid precipitation (or acidifying deposition) accelerates the decay rates of a wide variety of materials mainly because the presence of acids upon the material surfaces increases the flow across the surfaces of the electric currents that cause corrosion, discoloration, and embrittlement. Among the metals, ferro-alloys, copper, and some galvanized metals are known to be particularly susceptible. In some cases (e.g. zinc), the dissolution of the metal surface by acid precipitation is thought to increase the pH level of the product, thus resulting in an even more corrosive surface film. Carbonaceous building materials, such as limestone and cement, are more rapidly weathered, roughened, eroded, and stained. Paints are bleached and crystallized, and their drying and hardening times are increased. The tensile strength of textiles is degraded and textile dyes can suffer from fading. Losses of tensile strength also occur in paper, as does discoloration. Other cellulose products, such as wood, suffer similarly. Leather products deteriorate because the acids break down their fibrous structure. As Nraigu (1978) emphasizes, these processes are further intensified for those materials, such as cement, concrete, and some metals, often used in subaqueous and/or high temperature environments.

The recent economic impact studies of air pollution upon materials have yielded estimates for the entire United States of losses ranging from the \$904 x 10^6 Gillette (1975) attributed to sulfur oxides in 1968, to the \$3.8 x 10^9 Salmon (1972) attributed to all air pollution in 1970. None of these studies provides any substantial basis for attributing a portion of their estimated losses to acid precipitation, although the decline in sulfur dioxide levels throughout most of the eastern United States during the 1970's implies that an increased portion of whatever materials damages are occurring is attributable to acid precipitation. Most important, since all these studies basically do little more than inventory some existing materials, attach a market price to them, and then use physical science estimates of increases in replacement frequency to obtain a total loss estimate, they are susceptible to all the criticisms that can be directed toward most of the estimates In this chapter. As Glass (1978, p. 34) correctly points out, many extremely resistant materials, such as aluminum clad steels, have been widely adopted in the last decade; most estimates relate to uncoated rather than coated galvanized steels; the economic lives of many materials are so short (e.g., paper) that air pollution does not have time to affect them in a noticeable fashion; and, that when a substitute material. is adopted, the cost differential often cannot be assigned entirely to pollution since the substitute may have features that reduce cost dimensions other than useful life. In addition to these factors, the available studies sometimes have failed to discount the stream of costs properly. Moreover, all the studies have failed to consider that individuals may choose simply to bear a reduced stream of services from a material rather than purchasing a replacement, may alter behavior patterns so as to compensate for the stress that acid precipitation imposes upon the material, and may adopt materials more resistant to the ravages of acid precipitation. Finally, entire categories of useful materials such as limestone and concrete structures, including dams and pipes, and automobiles have had no economic attention devoted to them. Given the lack of economically useful physical science information and the lack of sound economic information, it is tempting to plead an absence of any basis whatsoever to make a judgment about either the fact or the potential for the economic impact of materials damages from acid precipitation and acidifying deposition. This is particularly so because many of the factors for which information is lacking can have either a positive or a negative economic impact. Of those factors that are most likely positive, or most likely negative, it is impossible to tell which will dominate. One is thus unable to state whether any estimate of the total (or marginal) impact represents an upper or a lower bound.

In spite of the preceding, it should be recognized that the costs of acid precipitation-induced materials decay could indeed be very substantial. The Commission on Natural Resources (1975, p. 696) refers to studies which estimated 1970 damages in Sweden to painted steels from all corrosion of $25.00 per capita (= $41.98 in 1978 dollars). From the same source, the Commission (197) quotes $23.00 per capita (= $38.64 in 1978 dollars) as being the total annual cost of deterioration of painted woodwork associated with all sources of deterioration. The studies from which these figures come appear to have at least as complete a physical science basis as any available, and no worse an economic basis than any of the extant studies.

Other than the direct and indirect products of chemical weathering, soiling is the major source of reductions in the usefulness of materials. Some of what passes for soiling (e.g., staining of the exterior stone surfaces of buildings) may, in fact, be chemically-induced discoloration. We thus presume that, in economic terms, soiling is relatively minor as opposed to chemical weathering. Some enhanced chemical weathering occurs to materials located near marine environments. This, however, is probably not an important source except for those materials frequently exposed to sea breezes and/or salt sprays. Some chemical weathering would naturally occur in humid areas since pristine precipitation is somewhat acidic (pH = 5.65). All these factors suggest that the aforementioned annual per capita costs in Sweden of the weathering from all sources of painted steels and painted woodwork are exaggerations of the 'losses caused by the impacts of acid precipitation upon these materials. However, as we previously emphasized, these materials constitute only a portion (though not a small portion) of the economically significant materials susceptible to acid precipitation-induced decay.

To generate a number for the materials damages caused by acid precipitation, we assume that the current per capita exposures of the great bulk of the Swedish population is very similar to the per capita exposures of the population in the eastern part of the United States. We further assume that the per capita mixes and magnitudes of painted steels and woodwork used by United States residents residing in Minnesota and east of the Mississippi River are similar to those of the Swedes. A simple multiplication of the sum of $41.98, for painted steels, and $38.64, for the painted woodwork, by the approximately 170×10^6 people residing in Minnesota and the states east of the Mississippi River in 1978, yields a calculated annual loss from materials damages of $\$13.71 \times 10^9$. It should be noted that this figure is an order of magnitude higher than previous estimates of all air pollution-induced materials damages over the entire United States. However, given both the physical science, economic, and inventory accounting limitations of the previous estimates (and this estimate), it seems as likely to be an underestimate and an overestimate. Nevertheless, given the difficulty and trivial gains to us in trying to justify the discrepancy between the above weak estimate for materials damages and those obtained by previous investigators, we do not deem this exercise to be a good forum for a display of intellectual stubbornness. We, therefore, state that materials damages are likely to be the largest category of the types of acid precipitation-induced damages we have surveyed, but we have no wish to assign to acid precipitation all materials damages that previous investigators have attributed to air pollution. We, therefore, set the 1978 materials damages caused by acid precipitation at $\$2 \times 10^9$, while recognizing that the figure could plausibly be much larger.

Summary and Conclusions

Although most of the analysis has been rather primitive, the economic benefits likely to accrue to a variety of life and property forms from the control of acid precipitation have been surveyed. It must be recognized, given the robust techniques available for doing economic assessments of the effects of acid precipitation, that the estimated magnitudes presented in this chapter cannot be justified indefinitely.

On the basis of our survey and synthesis of a fairly large volume of biological literature and stock, price, and output information, we conclude that if sufferers are viewed as either having to accept it or to take actions at their own expense to negate its effects, it is very unlikely that the current annual benefits of controlling acid precipitation for existing economic activities exceed \$5 $\times 10^9$ in 1978 dollars in Minnesota and the states east of the Mississippi River.[8/] Our best estimates are that \$2 $\times 10^9$ is in materials benefits, \$1.75 $\times 10^9$ is in forest ecosystem benefits, \$1 $\times 10^9$ is in direct agricultural benefits, \$0.25 $\times 10^9$ is in aquatic ecosystem benefits, and \$0.10 $\times 10^9$ is in other benefits, including health and water supply systems. The rationales supporting each of these sector estimates are presented in the chapter text. With the exception of a few instances where analogies could be drawn with the results of other studies using more robust estimation techniques, all these estimates disregard acid precipitation-induced price, activity, and location changes. We therefore have substantially more confidence in the rank-ordering by sector of the current annual benefits than we do in our estimates of the absolute magnitudes of these benefits.

If acid precipitation events continue to worsen, certain sectors could readily climb in the above ranking. For example, aquatic ecosystems currently have a relatively low position only because the geographical scope and severity of the aquatic acidification problem does not yet seem to be large enough to reduce substitution possibilities greatly across fresh-water fishing and hunting sites. Because of the water and soil treatment facilities already in place that can readily be adapted to handle liming procedures, the acid precipitation control benefits accruing due to the prevention of human health effects, indirect agricultural effects, and household, commercial and industrial water supply system effects are now and are likely to continue to be insignificant compared to the other classes of effects. Large-scale liming of aquatic and forest ecosystems appears to be neither technically or economically feasible.

The preceding conclusions are not the major conclusions we wish to draw from our survey and synthesis of the acid precipitation literature. We are unconvinced that either the above ordering or the above absolute magnitude estimates of the current annual benefits of control (even if correct) constitute the really important issues to consider when evaluating the acid precipitation problem. Indeed, we are unable to reject the discomforting notion that the effects for which one may feel secure using these simple or the much more sophisticated but still conventional methods of economic analysis reviewed in Chapter I are those having the least long-term economic significance. Instead, we suspect that these important issues relate to the impact of acid precipitation upon the stock and the assortment of natural resources. The next two chapters consider the implications of some of these issues regarding resource stocks and assortments for assessments of the benefits of controlling acid precipitation.

REFERENCES

1/ It should be noted that there may be some exceptions to acid precipitation acting as a sort of negative fertilizer. For example, Maugh (1979) reports on a TVA-sponsored study which found that if the sulfur emitted by coal- burning power plants in the Tennessee Valley region were removed, and not replaced by another sulfur source, crop production, especially cotton, would decline by at least 10 percent. Tisdale and Nelson (1976, p. 411) point out that raising soil pH in the Deep South to more than 6.0 may actually be harmful to yields.

2/ See Freeman (1.979) for a presentation of the conditions under which it would be a good approximation. For the Adams, et al. (1979) study, the on-farm value of the 14 crops was 16 percent less than the estimated sum of producer rents and consumer surpluses. When cotton was excluded the non- farm value of the 13 remaining crops was 20 percent less than the estimated sum of the producer and consumer surpluses.

3/ In a news item, Rich (1979) reports that field studies in southern Poland have attributed drops of 13 to 18 percent in the photosynthetic activity of pine needles subjected to wet and dry sulfur deposition. Dennis Knight of the Department of Botany at the University of Wyoming informs us that the Polish investigators believe that this reduction is due to SO_2 entering the leaf through the stomata and then being converted to H_2SO_4 <u>within</u> the leaf. This perspective may be contrasted with the bulk of the published literature which emphasizes the growth reducing properties of cuticular erosion and nutrient leaching from leaves and soil. Apparently, the Polish studies have not yet been widely distributed.

4/ In principle, the spreading of sufficient lime on <u>top</u> of forest soils might raise pH before precipitation moves down the soil column. Other than a vague statement by Rich (1979) on aerial lime spraying in Poland, we have found no commentaries on either the technical or the economic feasibility of this practice.

5/ If the forest growth effects of acid precipitation are viewed as analogous to a selective cutting policy, one could draw upon the technical forestry literature relating the effects of this type of cutting upon these components. We have not exploited the analogy here because of the likely wide variations in responses of the components to tree species mixes, topographical attributes, and other factors.

6/ Let V be the present value ($\$15.32 \times 10^9$) of the stream of losses, let A be the annual losses ($\$9.69 \times 10^9$), and let r be the rate of discount. Then:

$$V = A \left(\frac{1+r}{r}\right)[1 - (1+r)^{-\infty}].$$

The term in brackets can obviously be disregarded when one is dealing with an infinite future.

7/ According to the Economic Research Service (1.974, p. 37), 1970 self-supplied industrial water from saline sources in the United States was 10.07×10^9 gallons/day. It is unclear, however, how this use is distributed over ocean, estuary, and saline groundwater sources. The pH of any saline source could obviously differ greatly according to the extent to which the acidic fresh-water had been diluted by the saline water.

8/ In Hick's (1973) terms, the qualifying "if..." phrase indicates that an equivalent, as opposed to a compensating, measure of value is being employed. In effect, it is assumed that those who cause acid precipitation, rather than those who suffer from it, have the de facto property rights to the air resource. Moreover, since all our crude assessments are in willingness-to-pay terms, they will be less than if the assessment had been made in willingness-to-accept compensation terms [Randall and Stoll, (1980)].